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APPLYING HABITAT EQUIVALENCY ANALYSIS FOR CORAL REEF DAMAGE ASSESSMENT AND RESTORATION

J. Walter Milon and Richard E. Dodge

ABSTRACT

Quantifying economic damages and restoration measures for injuries to coral reefs has been a difficult task. In the U.S., habitat equivalency analysis (HEA) has emerged as a novel tool that combines biological and economic information to identify replacement habitats of an appropriate scale to substitute for the interim losses resulting from coral reef injuries. This article provides a review of the basic principles underlying HEA and a discussion of important considerations in applying HEA. These considerations include: how to describe coral reef functions and related human uses, recovery rates of coral reef organisms at injured sites with natural and active restoration, selection of replacement habitats and growth rates of organisms in these habitats, and the role of time and discount rates in the analysis. While HEA offers many advantages, specific decisions made in the application process can have a dramatic effect on the scale and cost of restoration and replacement habitat decisions. Management agencies and the scientific community need to be involved in developing standards for quantifying coral reef functions and recovery rates and the role of replacement projects in restoration planning.

Fishing and tourism activities that center on coral reefs provide the economic foundation for many local communities around the world. Widespread deterioration of coral reefs, due to both natural and anthropogenic causes, has led to growing interest in methods to assess damages and provide restorative measures (Precht, 1998). In the case of human induced damages such as vessel groundings, there is often a need to provide measures of the physical extent of the injuries and to determine economic compensation for the damages. While some efforts have been made to describe and quantify economic measures of coral reef values and damages (Hundloe, 1990; Finch et al., 1992; Julius et al., 1995a; Mattson and DeFoor, 1985; Spurgeon, 1992), the task has been difficult. As an alternative to direct economic measures, damage assessments in the U.S. have increasingly relied on a new method, habitat equivalency analysis (HEA), that combines biological and economic information to scale compensatory replacement projects for marine resource damages (Mazzotta et al., 1994; Unsworth and Bishop, 1994; National Oceanic and Atmospheric Administration, Damage Assessment and Restoration Program, 1995, 1997a).

In this article we provide a review of the basic principles underlying HEA and discuss important considerations for the parameter values used in applying HEA. The purpose of this review is not to evaluate the legal context for HEA since this may differ under various laws both in the U.S. and other countries. Instead, we seek to clarify some critical issues in the HEA methodology and encourage discussion about the role of HEA in coral reef damage assessment and restoration planning. We begin with a discussion of the relationship between coral reef functions and economic values and how changes in these functions due to human inflicted injuries can lead to economic damages. The role of restoration in mitigating on-site damages as well as providing off-site compensation for changes in coral reef functions is also addressed. In the third section we provide a detailed analysis of HEA focusing on the role of differences in coral reef functions and recovery rates in the methodology. We then turn to other components of HEA that require judgments about the suitability of off-site replacement projects and tradeoffs between current and future time periods. We conclude that HEA offers many significant advantages for coral reef damage assessment. Yet, given the novelty of the concept and the fact that most applications have been cloaked in the secrecy of litigation proceedings, the scientific community and the public need to be involved in developing standards for quantifying coral reef functions and recovery rates and the role of replacement projects in restoration planning.

Coral Reef Functions, Human Uses, and Measures of Economic Damages

The economic contributions of coral reefs to fishing, tourism, and other activities are generally well-known although few quantitative measures are available (Spurgeon, 1992; Hoagland et al., 1995). The relationship between economic activities and changes in coral reef functions due to human caused injuries, however, has received less attention. The basic premise for measuring economic damages is that an injury diminishes individuals' well-being (utility) through a reduction in the services provided by a natural resource (Kopp and Smith, 1993). For coral reefs, these services might include coral structures or fish that can be directly harvested or underwater landscapes that can be observed and/or photographed. For economic damages to occur, "some linkage must exist between the injury to the natural resource, the reduction in services, and the reductions in an individual's well-being" (Desvousges and Skahen, 1985: 1–7; Desvousges et al., 1997). A variety of methods have been developed to quantify economic damages depending on the specific linkage between the functions provided by the resource in the production of services to humans. A general framework for considering the linkages between coral reef functions, services, and economic measurement methods drawing from previous classifications developed by Smith and Krutilla (1982) and Desvousges and Skahen (1985) is presented in Table 1. Since the measurement methods are described in detail elsewhere (e.g., Freeman, 1993; Kopp and Smith, 1993), only the direct connections to coral reef assessments are discussed here.

The first grouping in Table 1 describes linkages in which changes in the economic value of coral reef functions can be measured through direct, indirect, or constructed human behavioral responses. Behavioral responses are the *sine qua non* for economic valuation (Bockstael and McConnell, 1999). For example, a direct linkage exists when damage to a coral reef reduces the availability of a marketable product such as ornamental pieces of coral or reef fish. Market price measures the per unit value of a lost product, so one measure of the loss could be derived by adding up the commercial values of all products that could not be harvested. It is important to recognize, however, that market prices also include costs related to harvesting that are not 'lost' if no harvesting occurs. Therefore, U.S. courts have generally defined damages for lost marketable products as the foregone profits (income) of harvesters (Jones et al., 1996). Because many countries prohibit harvesting from wild coral stocks, such a direct linkage is not likely to occur.

An indirect linkage implies that changes in coral reef functions effect some human activity such as recreational diving or fishing. The travel cost method evaluates changes in coral reef site usage after an incident to identify changes in users' visitation rates at the damaged site and other nearby sites. The foregone value of visits not made to the damaged site is the appropriate measure of damages (McConnell, 1993). The travel cost method could also be used to establish the economic value of a reef site prior to an incident. Then the percentage loss in reef area at a site could be viewed as an equal percentage reduction

Table 1. A taxonomy of linkages between coral reef functions, human uses and economic methods.

Linkage between resource functions and human use	Assumption for measurement	Economic measurement method
Behavioral		
Direct	Markets for resource products	Market prices/profits
Indirect	Resource functions provide use services for which markets exist	Travel cost Factor income/profits Benefit transfer
Constructed	Resource functions provide use services that can be described and valued by individuals	Contingent valuation Multiattribute utility
Nonbehavioral		
Direct	Functions provide use services in some nonquantifiable relationship	Restoration/replacement costs Habitat equivalency analysis
No linkage	No relationship between function and use services	Restoration/replacement costs Habitat equivalency analysis

in economic value. While this approach has been used in damage assessments (e.g., Julius et al., 1995a), it could be questioned since no explicit measure of lost use at the damaged site is derived. In some cases, damages could be measured by transferring a benefit (value) measure estimated in another travel cost study of a site that supported similar recreational activities. This benefits transfer method negates the need for a new valuation study of the injured reef site, but it raises additional questions about the quality and applicability of the transferred benefit measure (Freeman, 1995; Brouwer, 2000).

Another closely related indirect measure of damages is the change in income or profits for those who provide services to users of a damaged site. For example, charter dive boat operators and crew may suffer a decline in customers, and thereby income, due to damages at a coral reef site. This change in factor income (profits) represents a potential private claim of damages (Jones et al., 1996).

The third type of behavior based relationship presented in Table 1 is a constructed linkage in which carefully designed surveys are used to elicit measures of economic value for changes in natural resource functions. These surveys may use contingent valuation and/or contingent choice methods that ask individuals to directly or indirectly reveal their value (willingness to pay) for specific changes (Mitchell and Carson, 1989). These methods could be used with both users of a site and others who might have a value for that site so that both use and nonuse values may be elicited (Arrow et al., 1993). This approach to valuation has been controversial (e.g., Bjornstad and Kahn, 1996; Castle et al., 1994; Portney, 1994) and has not been used in the context of coral reef damage assessment.

The second grouping in Table 1 includes methods that may be useful when it is difficult or impossible to identify an explicit linkage between changes in coral reef functions and human activities. In this setting behavioral responses cannot be used to measure the actual economic value of losses resulting from coral reef damages. The alternative is to measure the costs of restoring and/or replacing the reef, but these costs may understate or overstate the actual economic losses. Where some direct but nonquantified relationship exists between coral reef functions and human activities, such as in the case of sport diving at an unmonitored site, restoration can be viewed as an effort to regain use of the site. Alternatively, if the injured site cannot be restored, a replacement site would support an equivalent type of activity. Either the restoration or the replacement provides compensation for the lost human uses provided by the injured site.

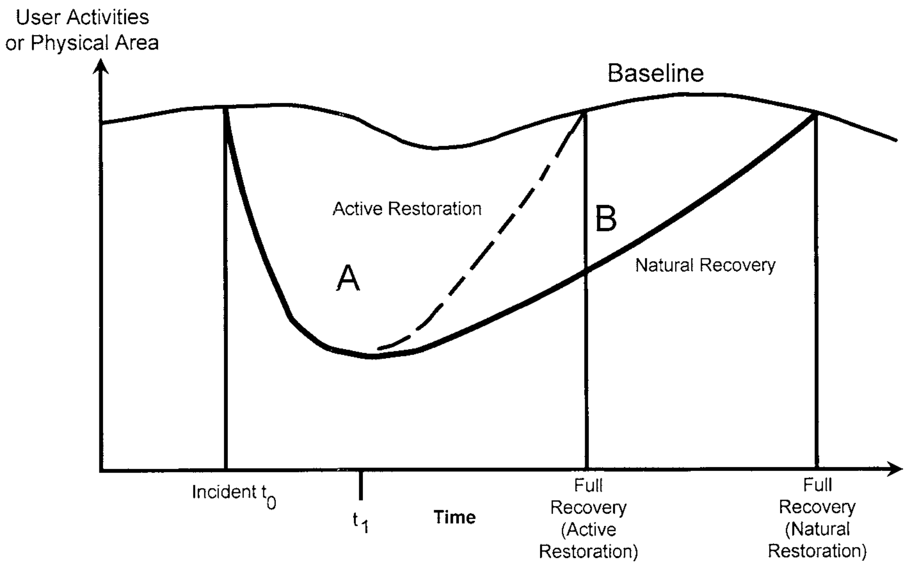

Figure 1. Relationship of natural recovery and active restoration to interim losses in user activities or physical area of coral reef habitat.

If there is no relationship between damaged coral reef functions and human activities, such as may occur with remote or unused reef sites, restoration or replacement only provides compensation for the biological and/or physical functions of the reef. In economic terms, this means that the impacted coral reef functions do not, directly or indirectly, effect individuals' well-being (utility). This distinction between compensation for lost human services versus compensation for lost ecological functions is important in considering the role of time in scaling replacement decisions. Moreover, it is important to recognize that because coral reef damages may not effect individuals' well-being does not imply that the total economic value of coral reef functions is zero. Aggregate values assigned to coral reef functions (e.g., Costanza et al., 1997; Peterson and Lubchenco, 1997) suggest the economic significance of these resources. But, these values are not linear functions of the area of the resource. A specific event may damage the biological functions of the resource yet leave the total value unchanged. This is analogous to 'threshold effects' in pollution damage models in which low levels of a pollutant have no effect on living receptors and thereby cause no change in economic value (Nichols, 1984).

Habitat equivalency analysis (HEA) is a method that has been used to determine the appropriate nature and scale of replacement projects resulting from vessel groundings on coral reefs (Florida Department of Environmental Protection, 1994; Julius et al., 1995b; National Oceanic and Atmospheric Administration, 1997b). To understand the role of HEA, it is useful to distinguish between the effects of restoration and replacement actions. Figure 1 represents time along the x-axis and either user activities or a physical unit metric, such as square meters (m^2) of reef surface area, along the y-axis. An initial, baseline level of ecological functions or user activities is provided by a coral reef habitat that would continue over time. At time t_0 an incident occurs resulting in a reduction in ecological functions of the habitat, and possibly, a reduction in human activities. Natural recovery of the site would result in a return of ecological functions/user activities along the solid curve depicted in Figure 1. With active restoration of the site at time t_1, the recovery path would progress along the dotted line in Figure 1. The loss of ecological functions/ human activities with natural recovery, represented by the area A plus B in Figure 1, would be reduced to area A alone with active restoration. The value of the quicker recovery in ecological functions/user activities is a measure of the benefits from active restoration. These benefits could be compared to the costs of active restoration.

Either with or without active restoration, Figure 1 shows that there will be some reduction in coral reef ecological functions or human activities caused by the incident. To compensate for these losses, it would be necessary to develop coral reef replacement habitat that provides biological functions or user activities equivalent to the losses (either area A with active restoration or area A plus B with natural recovery). HEA offers one approach to scale the coral reef replacement habitat by calculating the area necessary to replace the lost biological functions or user activities. And, based on the calculated area, the costs of providing replacement habitat is a (nonbehavioral) measure of the damages resulting from the incident. In most cases, it cannot be determined whether the replacement costs are less or greater than the economic value of the lost human activities. Therefore, the replacement costs should not be considered an acceptable substitute for utility-based (behavioral) measures of economic damage.

THE CONCEPTUAL BASIS FOR HABITAT EQUIVALENCY ANALYSIS

The basic logic of HEA is similar to the replacement ratio concept (Race and Fonseca, 1996) that has been used for wetland loss mitigation. A replacement ratio scales the size of the replacement project based on judgments about the importance of the injured habitat and the likelihood of success of a replacement (Fonseca, Julius and Kenworthy, 1999). For HEA, the replacement project is scaled to provide equivalent biological functions/ user activities for those lost during recovery from injury to a resource. A critical difference between HEA and a simple replacement ratio approach is that HEA requires specific assumptions about parameters in the model, thus selection of parameter values can have significant effects on the estimated replacement area.

To evaluate these effects, the lost biological functions/user activities described as either area A or A plus B in Figure 1 can be described as 'interim losses' (IL). Using a physical unit metric, m^2 of damaged reef surface area, IL can be expressed in terms of the cumulative lost reef surface area per year until recovery. Similarly, the 'replacement gains' (RG) provided by a replacement project can be expressed as the total incremental additions of reef surface area per year. Then, the size of the replacement project (R) in m^2 needed to provide an equivalent level of biological functions/user activities would satisfy the condition: IL = RG. Unlike the simple replacement ratio, the size of R depends on: a) the timing and duration of the recovery path for the injured site, and b) the timing and growth rate for the replacement habitat.

The parameters that govern this relationship can be described by defining IL in terms of the annual percentage reduction in damaged area at the injured site:

$$\sum_{t=0}^{t^*}\Big[\big((S - s_t/S) * S\big)\Big] \qquad \text{Eq. 1}$$

where S is the baseline level of functions/activities for the injured site measured in m^2, s_t is the regrowth (in m^2) at the injured site up to time period t, and t^* is the time period when the injured site is restored to baseline. The regrowth parameter, s_t, reduces the area that is damaged. For simplicity, it is assumed that the baseline is constant through time. Also, RG can be defined in terms of the annual percentage gain in area for the replacement project:

$$\sum_{t=t_1}^{T}\left[(r_t/R)*R\right] \qquad \text{Eq. 2}$$

where R is the size (in m^2) of the replacement project, r_t is the growth (in m^2) of the reef functions/activities at the replacement, t_i is the time period when the replacement project is established, and T is the terminal time in the planning period. In this specification, the growth parameter, r_t, increases the area of the replacement habitat that provides equivalent reef functions. By setting (1) equal to (2) and solving for R, the size of the reef replacement project can be determined. In general, longer recovery periods for the injured site will require more replacement habitat.

This basic approach to establishing habitat equivalency is useful for uniform landscapes with little difference in biological functions across the injured area (Mazzotta et al., 1994). In the coral reef setting, this approach may not realistically account for the diverse assemblage of organisms within the injured area and differences in regrowth/growth of these organisms at the injured and replacement sites. A more general form of the HEA that accounts for different reef organism populations can be developed by redefining IL as:

$$\sum_{k=1}^{K}\sum_{t=0}^{t^*}\left[(S_k - s_{kt}/S_k)*S_k\right] \qquad \text{Eq. 3}$$

where S_k is now the baseline area (in m^2) of injury to the k^{th} reef organism population (k = 1, ..., K), s_{kt} is the regrowth (in m^2) of the k^{th} reef organism population at the injured site through time period t, and t^* is the time period when the k^{th} injured organism recovers to baseline. A similar redefinition of RG is:

$$\sum_{k=1}^{K}\sum_{t=t_i}^{T}\left[(r_{kt}/R_k)*R_k\right] \qquad \text{Eq. 4}$$

where R_k is the size (in m^2) of the replacement project for the k^{th} organism, r_{kt} is the growth (in m^2) of the k^{th} organism at the replacement, t_i is the time period when the replacement project is established, and T is the terminal point for the analysis. Note that each of the k^{th} organisms may return to the baseline level at different times in both (Eq. 3) and (Eq. 4). In this general population form, each of the k areas would be solved for the R_k that provides an equivalent amount of the reef organism over time and then the individual k areas would be summed to determine the total area for the replacement project.

Table 2. Replacement habitat area[1] for basic and general HEA with alternative planning periods.

Reef organism	35 yr period		100 yr period	
	Landscape HEA	Population HEA	Landscape HEA	Population HEA
Stony corals	947	284	213	64
Gorgonian corals	—	80	—	25
Algae	—	31	—	11
Total replacement area (m^2)	947	395	213	100

[1]Based on 1000 m^2 of injured area with a replacement project beginning the same year as the injury; injured area composed of 30% stony corals, 30% gorgonian corals, and 40% algae and other encrusting organisms. Calculated values rounded to whole numbers.

The differences between the basic landscape and more general population forms of the HEA-based approach can be considered with a numerical example. This example is intended solely to illustrate HEA-based analysis and the parameter values have been selected to illustrate specific issues that may arise in an actual application. Assume a coral reef surface area of 1000 m^2 is injured and the organism populations are: 30% (300 m^2) stony corals, 30% (300 m^2) gorgonian corals, and 40% (400 m^2) algae and other encrusting organisms. For this example, we assume no active restoration to offset interim losses and natural recovery rates to baseline as follows: stony corals – 35 yrs, gorgonian corals – 15 yrs, and algae/other organisms – 5 yrs. Regrowth of all organisms begins immediately after the injury and is linear (constant percentage gain). A replacement habitat (such as an artificial reef module) will be established near the injury site in the same year as the injury. For simplicity it is assumed that each organism population will colonize and grow on the replacement habitat in the same composition and in the same time period as the injured site. Two planning horizons, 35 and 100 yrs, are also included to illustrate the effects of changes in this component of the HEA. For the landscape HEA (Eq. 1 and Eq. 2) it is assumed that the reef landscape would be classified as a 'stony coral landscape' so the recovery time for the entire 1000 m^2 area would be 35 yrs. With the population HEA (Eq. 3 and Eq. 4), the respective areas and faster recovery/growth rates for the gorgonian and algae populations would be included in the analysis.

The resulting estimates of replacement habitat area calculated with the landscape and population HEA are reported in Table 2. First, the landscape HEA results in significantly larger replacement area-equivalents because the longest recovery/regrowth period was used to characterize the habitat. With the shorter recovery times for gorgonian corals and algae/other organisms included in the population HEA, the size of the replacement habitat is reduced. This implies that characterizing the landscape in terms of the longest recovery time for the reef organisms provides an upper bound for the interim losses. Second, extension of the planning period from 35 to 100 yrs results in a significant decrease in the required replacement habitat in both the landscape and population HEA. This occurs because the longer planning horizon allows additional amounts of replacement habitat to accrue once the injured area(s) is (are) recovered and the replacement area is fully functional. Finally, note that the HEA area-equivalents from these parameter values are all less than the original injured area.

These large differences between the landscape and population versions of HEA are partially attributable to the decision to represent recovery of the damaged areas in the landscape HEA with the stony coral regrowth rate of 35 yrs. This would be a conservative approach that allowed for the slowest growing organism to recover but it would underestimate regrowth of other populations. Assume as an alternative that the gorgonian coral recovery/growth rate of 15 yrs was used in the landscape HEA. With 35 and 100 yr planning periods, 1000 m^2 of injury to a gorgonian coral landscape would require 265 m^2 and 81 m^2 of replacement habitat, respectively. These estimates are now smaller than the population HEA results in Table 2 indicating that the decision how to characterize the injured coral reef habitat is not a trivial issue in HEA.

These examples also help to illustrate the important role of time in the HEA framework. But, this role extends beyond the choice of the planning period. Since coral reef organisms in the injured and replacement habitats will recover and become functional at different times in the future, it is pertinent to question whether the area-equivalents should be 'discounted'. In economic terms, discounting introduces the 'price of time' that reflects ethical beliefs about preferences for the present and future (D'Arge, 1993; Heal, 1998). Typically discounting is applied to future economic values, such as the dollar value of an annuity received at some date in the future, and the discount rate converts the future value received into its present value. Higher discount rates imply stronger preferences for present consumption; discount rates can vary across countries due to differences in economic conditions and public preferences (Lind, 1990). In the context of the landscape HEA, the 'discounted' equivalence between IL and RG could be rewritten as:

$$\sum_{t=0}^{t^*} D_t\left[(S - s_t/S) * S\right] = \sum_{t=t_i}^{T} D_t\left[(r_t/R) * R\right] \qquad \text{Eq. 5}$$

where $D_t = (1 + d)^{-t}$ and d is the discount rate. A comparable expression could also be developed for a discounted version of the population HEA using equations (3) and (4).

The decision whether discounting is applicable in HEA depends on what is being replaced. If the analysis is intended to evaluate a direct (non-behavioral) linkage between biological function and human uses of a coral reef (Table 1), then the loss of human use indicates a loss of economic value in future periods. A discount rate is necessary to adjust for differences in the value of human use during recovery of the injured area and growth of the replacement. This assumption of a direct relationship between habitat functions and human use is implicit in the theoretical frameworks for HEA developed by Mazzotta et al. (1994) and Unsworth and Bishop (1994). On the other hand, if the injury results in a loss of biological function with no lost human use, discounting would not be appropriate since there is no lost economic value. In economic terms, this means that the change in the total value of the habitat (the marginal value) due to the injury is zero. This does not imply that the habitat has no economic value, but only that the injury does not change the total value. Other reasons have also been advanced for a 0% discount rate in economic analysis of long term projects (even with a change in total value) based on concerns about intergenerational equity (Howarth and Norgaard, 1993) and sustainability (Heal, 1998).

To illustrate the effects of discounting on the scaling of replacement habitat in the alternative HEA frameworks, representative discount rates of 0, 3 and 6% were combined with the information and assumptions given above for the examples presented in Table 2. Then, estimates of replacement habitat from the landscape and population versions of HEA were calculated for six alternative planning periods. Longer time periods represent

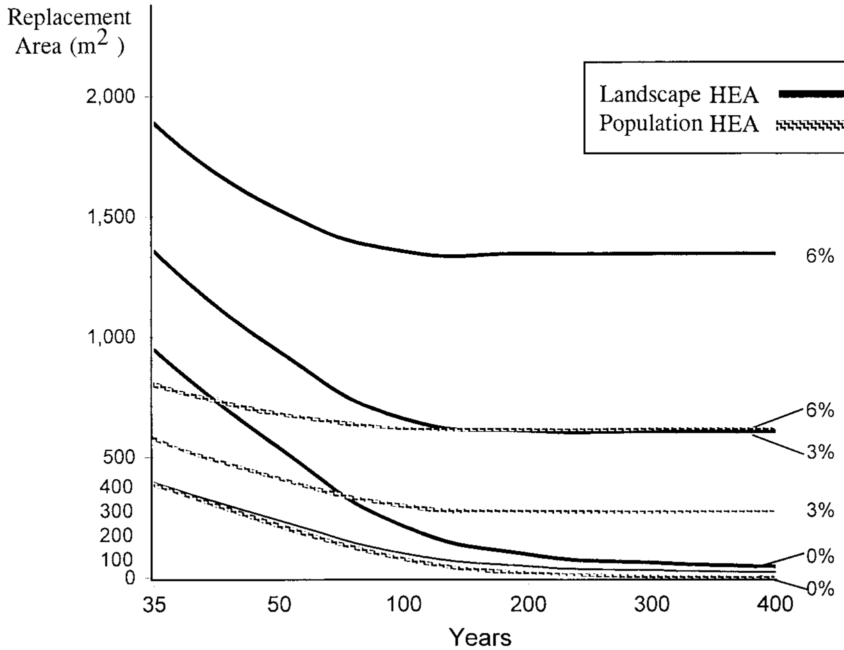

Figure 2. Replacement areas for landscape and population HEAs with alternative discount rates and planning periods.

the concept of 'perpetuity' that has been used in previous applications of HEA (e.g., Julius et al., 1995b; Julius, 1998).

Figure 2 shows that the choice of discount rate has a dramatic effect on HEA results. Estimated replacement areas vary from nearly double the original area of injury to less than 10% of the original area. Higher discount rates produce larger estimates of required replacement areas regardless of the version of HEA. This occurs because the loss of reef habitat during the more immediate future receives greater weighting than the growth of replacement habitat in the more distant future. With a 6% discount rate, the landscape HEA produces required replacement areas that are always greater than the original area of injury (1000 m^2). But, the replacement area results from the population HEA with a 6% discount rate are all less than the injured area indicating the wide variability that can occur with different analytical assumptions.

Similarly, the landscape HEA with a 3% discount rate produces some estimates of replacement area greater than the injured area, but these estimates converge rapidly once the planning period exceeds 100 yrs. This is generally true with discount rates greater than 0 because positive discount rates effectively nullify any values beyond 100 yrs. Since a 0 percent discount rate weights each time period equally, no discounting effect occurs as the time period increases. Overall the results in Figure 2 demonstrate that, even with the same information/assumptions about the injured reef organisms, the combination of alternative HEA frameworks coupled with various discount rates can lead to significantly different estimates of required replacement habitat in a coral reef damage assessment. Unlike the traditional replacement ratio concept used in other damage mitigation settings where the replacement area has typically been greater than the injured area (Race and Fonseca, 1996), the replacement area estimated from HEA may be greater than, equal to, or less than the injured area depending on the choice of parameter values.

Other Considerations

The preceding examples neglected the role of active restoration in mitigating the interim losses that may result from injury to a coral reef habitat. The HEA-based approach does not directly address the scale or cost of active restoration, yet results from HEA are directly dependent on changes in the timing and extent of interim losses. In principle, active restoration should be undertaken only if it reduces the interim losses since resources devoted to active restoration could be diverted to creating more replacement habitat. The issue, however, is deciding how large the reduction in interim losses of the coral reef habitat would be with different types of active restoration and how the costs of this restoration compare to the costs of replacement habitat.

To illustrate this problem, suppose that recovery of injured stony corals in the example presented earlier in Table 2 would occur in 25 yrs with active restoration instead of 35 yrs with natural recovery. Using a landscape HEA with a 35 yr growth rate for the replacement habitat, a 35 yr planning period, and a 0% discount rate, the required replacement area would decrease to 659 m^2 (from 947 m^2 in Table 2). Suppose the cost of active restoration was \$200,000 and the cost of replacement habitat was \$1000 m^{-2}. Active restoration is a sound decision since the 'savings' from a \$200,000 investment in restoration is \$88,000 (288 m^2 times \$1,000 m^{-2} minus \$200,000). On the other hand, if the cost of replacement habitat was \$200 m^{-2}, active restoration should be carefully scrutinized since 1000 m^2 of replacement habitat could be created for the same cost. Thus, the selection of active restoration and replacement habitat options from HEA is a joint decision.

A closely related consideration is the suitability of replacement habitats to compensate for the resource functions/human uses lost due to coral reef injuries. The preceding discussion of alternative HEA frameworks and examples has utilized surface area as the common metric to equivalate interim losses (IL) and replacement gains (RG). This metric has distinct advantages for evaluating the biological functions of injured coral reefs since surface area of the landscape or organism populations can be measured at an injury site. More problematic, however, is the use of surface area as a metric for the biological functions of a replacement habitat. These habitats might vary from engineered artificial reef structures to surplus materials such as concrete rubble or derelict vessels. While a variety of materials have been used as artificial reefs to achieve numerous objectives, few studies have documented the success of these materials in replacing specific coral reef functions (Miller and Falace, 2000). Certainly the working hypothesis (used in the HEA examples above) that coral reef organisms would colonize and grow on the replacement habitat in the same composition and at the same rates as a natural reef site has not been tested in practice. Moreover, the costs of specific types of replacement habitat may vary widely. A recent review conducted by the authors revealed that the acquisition costs of engineered artificial reef modules manufactured by U.S. companies ranged from \$150 m^{-2} to over \$2000 m^{-2}.

These difficulties with the use of surface area as a scaling metric in HEA for coral reef injuries are further amplified if the purpose is to replace lost human uses. Recreational uses of coral reefs such as fishing or diving are typically measured in trips or user-occasions. Occasionally these measures have been converted to metrics such as trips per unit area of reef as an indicator of a linkage between ecological functions and human uses (e.g., Dixon et al., 1993). But, providing replacement surface area is no guarantee that trips would actually occur because human uses may respond to perceived quality differences between sites. While a variety of techniques exist to identify and evaluate the effects of quality differences on various human uses of natural resources (e.g., National Oceanic and Atmospheric Administration, 1997b), no studies currently exist for coral reefs and the suite of options for replacement habitats.

Conclusions

Quantifying economic damages resulting from injuries to coral reefs has been a difficult task. While a number of alternative methods can be used, identifying specific linkages between lost reef functions and human activities is costly, time-consuming, and rarely attempted in practice. In the U.S., habitat equivalency analysis has emerged as a novel tool to identify replacement habitats of an appropriate scale to substitute for the interim losses resulting from coral reef injuries. While the primary focus to-date for HEA-based applications has been coral reef injuries from vessel groundings, the method is adaptable to other discrete events that might cause injuries to coral reefs such as beach renourishment and port development projects.

In this article we have discussed a number of issues that must be considered in any application of HEA for coral reef damage assessment and restoration planning. These issues can be summarized as:

- Whether to characterize the injured reef area as a uniform landscape or as an assemblage of organisms with varying growth/regrowth rates;
- Whether the interim loss of biological functions from an injury also results in loss of human uses;
- The choice of an appropriate metric to represent interim loss of biological functions and/or human uses;
- The selection of an appropriate replacement project to substitute for interim loss of biological functions and/or human uses;
- The selection of recovery/growth rates for the reef landscape/organism populations at the injured and replacement sites;
- The choice of planning periods for the analysis; and,
- The choice of an appropriate discount rate if the injury results in a loss of economic value attributable to the coral reef habitat.

As demonstrated above, specific decisions about these parameters for HEA can have a dramatic effect on the scale and cost of injury site restoration and replacement habitat decisions. While these decisions will inevitably be made within the context of specific injury events and under different legal authorities, it is unlikely that these decisions will achieve desirable long-term results unless they are informed by more research and dialogue among physical scientists, economists, and stakeholders involved in coral reef damage assessment and restoration planning. Growing concerns about the effects of wetlands loss mitigation through the creation of wetland habitat (e.g., Roberts, 1993) should provide a note of caution to those involved in coral reef replacement decisions.

The necessary research agenda to address these issues is lengthy, but several items are clear priorities. First, long-term monitoring of natural and active recovery on damaged coral reef sites is necessary to establish meaningful estimates for interim losses of biological functions. Similarly, the deployment and monitoring of replacement habitats must be conducted to test specific hypotheses about reef organism recruitment, growth, and long-term success in replacing coral reef functions. Third, more effort must be given to document losses/changes in human uses and economic values at damaged coral reef sites and at replacement habitat sites. Imprecise references to human impacts from coral reef injuries can quickly become controversial and hinder the search for appropriate replacement projects. Finally, it is likely that specific HEA studies conducted for legal proceedings will be viewed as arbitrary until government agencies responsible for coral reef damage assessment and restoration planning provide guidelines and generally acceptable recovery/growth rates for coral reef organisms. Ideally this information would evolve through a notification and public comment process so that scientists and others involved in assessment and restoration planning have the opportunity to share research results and experience.

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